

1 Improving estimates of tropical peatland area, carbon storage, 2 and greenhouse gas fluxes

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46 **Abstract**

47 Our limited knowledge of the size of the carbon pool and exchange fluxes in forested lowland tropical
48 peatlands represents a major gap in our understanding of the global carbon cycle. Peat deposits in
49 several regions (e.g. the Congo Basin, much of Amazonia) are only just beginning to be mapped and
50 characterised. Here we consider the extent to which methodological improvements and improved
51 coordination between researchers could help to fill this gap. We review the literature on measurement
52 of the key parameters required to calculate carbon pools and fluxes, including peatland area, peat bulk
53 density, carbon concentration, above-ground carbon stocks, litter inputs to the peat, gaseous carbon
54 exchange, and waterborne carbon fluxes. We identify areas where further research and better

coordination are particularly needed in order to reduce the uncertainties in estimates of tropical peatland carbon pools and fluxes, thereby facilitating better-informed management of these exceptionally carbon-rich ecosystems.

Keywords

Peat, greenhouse gases, remote sensing, tropical ecology, carbon cycle.

I Introduction

Most peat in the tropics is located in the lowland humid forests of Southeast Asia, Amazonia, Central America and equatorial Africa (Figure 1). Page et al. (2011b) estimated the extent of tropical peatlands at 441,000 km² and their carbon (C) pool at 81.7–91.9 Gt C. This carbon pool is susceptible to climate change as well as local human impacts. Some peatlands in inland regions of Kalimantan have lost carbon due to increasing El Niño intensity and changing sea levels during the late Holocene (Dommain et al. 2011, 2014), suggesting that these peatlands may also respond to future (anthropogenic) climatic change. More recently, peatlands throughout Southeast Asia have been degraded by logging and plantation development (Miettinen et al. 2012). Drainage to improve growing conditions for crops such as oil palm and *Acacia* (for pulpwood) leads to peat subsidence and enhanced CO₂ emissions persisting over decades (Jauhiainen et al. 2012; Hooijer et al. 2012). Accidental burning, due mainly to small-scale land clearance fires getting out of control, can lead to very large peat losses over weeks or months (e.g. in the exceptional El Niño year of 1997, CO₂ emissions from Southeast Asian peatland fires were equivalent to 13–40% of total global fossil fuel emissions: Page et al. 2002). Crucially, whilst a large proportion of lost tropical forest biomass can be recovered within decades in secondary-growth forest (Letcher and Chazdon 2009), restoration of peatland carbon stocks to pre-disturbance levels would take thousands of years.

Despite their importance, basic information on carbon storage in tropical peatlands is lacking. This is not just a problem in the tropics. Reviewing estimates of the size of the boreal peatland carbon pool, Vasander and Kettunen (2006) showed that they vary by an order of magnitude (from 41.5 Pg C for all

80 histosols globally: Buringh 1984, to 455 Pg C for boreal and subarctic peatlands: Gorham 1991) due
81 mainly to substantial variation in estimates of peat dry bulk density (DBD) and thickness. The same
82 problem applies in the tropics but with the additional difficulty that, unlike in boreal peatlands, the
83 area of peat is also very poorly known in some regions.

84 Most research in the tropics has focused on Southeast Asian peatlands, and their distribution is
85 comparatively well known, yet large uncertainties still exist in estimates of the amount of carbon
86 stored in them. For example, published figures for Indonesia include 37.2 (Wahyunto et al. 2003, 2004,
87 2006), 55 ± 10 (Jaenicke et al. 2008), and 57.4–58.3 Pg C (Page et al. 2011b), based on differing
88 methodologies. New field data can make a substantial difference: Dommain et al. (2014) estimated the
89 peatland carbon pool in western Indonesia (excluding Papua) at 23.1 Pg C, a substantial downward
90 revision of the 33.3 Pg C estimate for the same area by Wahyunto et al. (2003, 2004) due mainly to
91 new data on DBD and peat thickness, even though the area of peat estimated by the two studies (which
92 are based on similar GIS datasets) was almost identical (131 500 and 129 700 km² respectively).
93 Peatlands in other tropical regions are less well mapped. For example, although wetlands in the
94 Cuvette Centrale of the Congo Basin, the fourth- or fifth-largest wetland on Earth (Campbell 2005),
95 have been mapped using remote sensing techniques (e.g. Bwangoy et al. 2010; Betbeder et al. 2014),
96 the extent of peat within these wetlands is essentially unknown; published estimates of the size of the
97 peatland carbon pool in the Congo basin are based on very few field data (Joosten et al. 2012). In
98 Amazonia, the existence of c. 40 000 km² of peat in the Pastaza-Marañón basin, Peru, has only recently
99 been confirmed by fieldwork (Lähteenoja et al. 2009a,b, 2012). The uncertainty in this peat carbon
100 pool is very large (0.8–9.5 Gt: Lähteenoja et al. 2012). Limited surveys have been carried out in parts of
101 Brazil (Lähteenoja et al. 2013) and French Guiana (Cubizolle et al. 2013), but remain to be conducted
102 elsewhere in the Amazon basin. A substantial revision of the global estimate of the size of the tropical
103 peatland carbon pool is therefore likely over the next few years. Data on carbon fluxes, especially of
104 the important greenhouse gas (GHG) methane, are likewise extremely scarce, especially outside
105 Southeast Asia.

106 The high carbon density of tropical peatlands makes them an obvious focus for emissions-
107 management schemes such as the UN REDD+ (United Nations Reducing Emissions from Deforestation
108 and Forest Degradation) programme (Murdiyarso et al. 2010). This will require precise inventories of
109 peatland carbon stocks and fluxes to be carried out for specific regions in order to provide accurate
110 estimates of baseline carbon pools and fluxes against which projected reductions in carbon emissions
111 can be measured, verified, and translated into carbon credits.

112 Therefore two fundamental research priorities are (1) to quantify the amount of carbon stored in
113 tropical peatlands accurately at a range of scales, and (2) to quantify fluxes of carbon to and from the
114 tropical peatland carbon pool. More broadly, a better understanding of the present distribution of
115 tropical peatlands, the processes of peat accumulation and decay, and the development of peatlands
116 over time is relevant to other important research questions, including:

- 117 • What determines the distribution of peat in the tropics?
- 118 • How will tropical peat stocks change in the future?
- 119 • How do tropical peatlands influence biogeography and biodiversity?
- 120 • How should tropical peatlands be managed?

121 In this paper we aim to encourage better and more consistent methodologies for producing carbon
122 inventories and budgets, principally at a regional scale. The value of a coordinated approach is clear:
123 for example, where long term, repeated and systematic carbon inventories of tropical forest biomass
124 have been coordinated by the RAINFOR (Malhi et al. 2002) and AFRITRON networks (among others),
125 many new insights into long- (Baker et al. 2004) and short-term (Lewis et al. 2011) carbon dynamics
126 have emerged, generating over 100 carbon-focused publications by more than 200 collaborators.

127 The paper is divided into three parts, 1) mapping peat distribution, 2) estimating the size of the carbon
128 pool, and 3) estimating carbon fluxes. In part 1, we focus on emerging remote sensing technologies
129 which, alongside appropriate measurements on the ground, can improve our ability to map peatland
130 extent. In part 2 we discuss the measurement of peat thickness, peat bulk density and carbon
131 concentration. In part 3 we discuss carbon fluxes into and out of peatland ecosystems, specifically,

132 long-term carbon accumulation rates, litter inputs and decomposition, gaseous and waterborne fluxes,
133 and above-ground carbon stocks. We focus on forested tropical lowland peatlands, although we
134 recognize that many of the issues we discuss also pertain to other types of peatland in the tropics and
135 beyond.

136 **II Mapping peat distribution**

137 Field mapping of peatlands is a considerable challenge, especially at regional, national and global
138 scales. However, a combination of field measurements and inferences from remote sensing can
139 provide an optimal balance, where realistic programmes of fieldwork can yield map-based products
140 that cover a region comprehensively, are reasonably accurate and reliable, and with robustly
141 quantified uncertainties.

142 Tropical peatlands are often distinct from surrounding *terra firme* (dry-land) forests in four ways that
143 are observable in satellite and airborne data. Firstly, their vegetation is often low in diversity. In South
144 and Central America, some parts of Africa, and on the island of New Guinea, palms are often more
145 abundant than in upland forests, sometimes forming mono-dominant stands (e.g. Lahteenoja and Page
146 2011; Wright et al. 2011). However, diversity in some peatland forests can be high (e.g. Sumatran
147 swamps: Brady 1997). Secondly, vegetation structure is often (but not always) distinctive, often with
148 more open canopies and low-stature or thin-stemmed trees, or no trees at all (e.g. Anderson 1983;
149 Page et al. 1999; Phillips et al. 1997). Thirdly, their topography can be distinctive. Tropical peatlands
150 typically occupy a specific topographic or geological setting, for example the subsiding Pastaza-
151 Maraon basin in Peru or the peats forming along dendritic drainage channels in the Tasek Bera basin,
152 Malaysia (Wust and Bustin 2004); blanket peats that are indifferent to topography only occur rarely in
153 upland settings (Gallego-Sala and Prentice 2012). Many tropical peatlands are also detectably dome-
154 shaped (e.g. Phillips et al. 1997; Jaenicke et al. 2008; Lahteenoja et al. 2009b). Finally, peatland water
155 tables often lie close to, at or above the surface throughout the year (e.g. Lawson et al. 2014). Whilst
156 any one of these four features alone is insufficient to characterise an area of forest as potentially peat-
157 forming, the combination of two or more presents a much stronger case (Draper et al. in press).

158 These properties can be mapped using a number of different remote sensing products. Compositional
159 and structural features of peatland vegetation have been distinguished using optical sensors such as
160 Landsat (Phua et al. 2007; Jaenicke et al. 2010; Li et al. 2010; Lahteenoja and Page 2011), SPOT
161 (Systeme Pour l'Observation de la Terre; Lee 2000; Miettinen and Liew 2010), and MODIS (Moderate
162 Imaging Spectrometer; Langner et al. 2007; Wijedasa et al. 2012). Figure 2 presents an example of
163 vegetation classification of the Changuinola peat dome in San San Pond Sak, Panama, using multi-scale
164 Landsat Thematic Mapper (TM) image analysis supported by aerial photography and field data to
165 characterise the main vegetation gradient. To date, optical imagery from medium spatial resolution
166 sensors such as the Landsat series (30 m multispectral imagery) has been the primary and most
167 successful tool for mapping peatlands. The new generation of VHR (very high resolution) products,
168 such as IKONOS (4 m multispectral) or WorldView-2 (2 m multispectral) imagery, potentially brings
169 new opportunities for detailed and accurate vegetation mapping. One key difficulty in the tropics is the
170 infrequent temporal coverage of these sensors, which makes cloud-free images difficult to obtain.
171 RapidEye (5 m multispectral) products, derived from a constellation of five satellites with
172 consequently more frequent image acquisition, can be a more reliable source of cloud-free imagery
173 (e.g. Franke et al. 2012). Opportunities are likely to grow further as new sensors are launched (e.g.
174 1.24 m resolution WorldView-3) and future missions become operational (e.g. the EU's Sentinel-2 10
175 m resolution twin satellites),

176 Active sensors such as radar and LiDAR (Light Detection And Ranging) which penetrate the canopy
177 can be used to detect the distinctive forest structures (e.g. combination of low canopy, thin stems, high
178 stem density in pole forest) or patterns in structure (e.g. concentric zonation of vegetation
179 communities) that characterize some peatlands. Radar and LiDAR are also able to provide topographic
180 data which can help to distinguish between peatland and *terra firme* forests, but few attempts have
181 been made to identify tropical peatlands using these tools. One useful exception is the use of an orbital
182 LiDAR instrument onboard the ICESAT satellite to measure peat topography and forest biomass at a
183 study site in Kalimantan (Ballhorn et al. 2011; other examples include Hoekman and Vissers 2007;
184 Rakwatin et al. 2009; Jaenicke et al. 2008; Jubanski et al. 2013). A limitation of LiDAR products is that
185 they are generally available as discrete point measurements (orbital sensors) or thin strips of data

186 (aerial sensors) rather than a full coverage. Other data sources (e.g. L-band radar data and ground-
187 based measurements of forest structure) are usually needed to interpolate between LiDAR
188 measurements (Mitchard et al. 2012).

189 The presence of standing water below a forest canopy can produce a distinctive radar backscatter
190 signal, particularly at longer radar wavelengths. L-band radar has been used extensively in tropical
191 contexts to map standing water (de Grandi et al. 1998, 2000; Hess et al. 2003; Hoekman 2007;
192 Bwangoy et al. 2010), to track changes in floodwater extent (Rosenqvist and Birkett 2002; Alsdorf
193 2003; Jung et al. 2010; Lee et al. 2011; Betbeder et al. 2014), and, using time series of radar data, to
194 distinguish areas that are constantly wet from those that are only seasonally wet (Waldram 2014).

195 Remote sensing therefore already provides effective tools for extrapolating from field measurements
196 to map peatlands over large areas, but there is scope for further methodological research. The use of
197 optical sensors to define peatland extent has been widely and successfully implemented, but the less
198 commonly used active sensors may discriminate between peatland and *terra firme* more effectively in
199 many circumstances. We recommend that wherever possible, multiple remote sensing products
200 (including both optical and active sensors) should be used in combination. Some of the data types
201 discussed here, such as airborne LiDAR, are too expensive to obtain for many projects. However,
202 suitable products for mapping peatlands based on vegetation structure and composition, topography
203 and inundation at a coarse scale (c. 30 m) are freely available from Landsat, SRTM (Shuttle Radar
204 Topography Mission) and ALOS PALSAR. A combined approach using these products is a highly
205 feasible starting point for future mapping projects.

206 It remains the case, however, that interpretations of peatland distribution based on remote sensing
207 need to be validated with ground reference data. Ground reference points are often difficult and
208 expensive to install and therefore require careful planning in order to maximise their usefulness. They
209 need to meet many criteria: they should be widely distributed across the study region, encompassing
210 the full range of spatial heterogeneity; they need to span the full range of environments present in the
211 landscape (including anthropogenic ones such as rice fields and oil-palm plantations: Miettinen et al.
212 2012), not just those associated with peat; and the reference dataset should be sufficiently large both

213 to develop and validate a classification using different subsets of the data. Typically, hundreds of
214 points are required to develop a reliable classification for a particular region, so it would clearly be
215 desirable if researchers visiting new sites could collaborate in generating suitable data. In Table 1 we
216 recommend a set of measurements that are needed for synoptic mapping and ground reference, and
217 which can easily and quickly be collected as part of the basic site description for any kind of research
218 on tropical peatlands. More extensive and specialized measurements (e.g. detailed vegetation
219 composition, water table depth variation) may be necessary for more specific applications of remote
220 sensing, but wherever possible they should be carried out in a way that preserves the compatibility of
221 measurements between studies, in line with Table 1.

222 **III Carbon stocks**

223 The quantity of carbon stored by peatlands in a region (M_c , in kg) can be calculated as:

$$224 \quad M_c = AD\rho c \quad (1)$$

225 where A is the area of peatland (m^2), D represents its mean thickness (m), ρ is the mean DBD (kg m^{-3})
226 and c is its carbon concentration (dry mass proportion; Gorham 1991). The area of peatland can be
227 determined by a combination of remote sensing and ground survey, as discussed in Section II. The
228 remaining three variables on the right hand side of equation (1) are each susceptible to considerable
229 uncertainty, which in combination can lead to very large uncertainties in M_c .

230 *1 Peat thickness*

231 In carbon inventory research it is usually convenient to treat peat as a separate, particularly carbon-
232 rich category of soil, but a long-standing problem in peat research, unlikely to be resolved any time
233 soon, is how to define 'peat'. Most workers define a peat soil as one that contains more than a certain
234 proportion of organic matter, but the critical value varies widely, between about 30 and 65 wt%
235 organic matter (Joosten and Clark 2002:41; Wüst et al. 2003), hindering data synthesis (Page et al.
236 2011b). Some peat units have a clearly-defined contact with the underlying, less organic material, but
237 others change in composition more gradually. In such cases it can be impossible to judge peat

238 thickness consistently in the field, and unfortunately, core samples are not always taken for laboratory
239 analysis of organic matter content, resulting in unreliable data. Researchers must acknowledge that
240 'peat' is *de facto* a flexible term, and circumvent definitional issues by collecting objective data on the
241 properties of the material they are studying. One way forward, and our recommendation, is to build on
242 past efforts such as the CARBOPEAT project (<http://www.geog.le.ac.uk/carbopeat>) to compile the
243 necessary data (core location, sample depth, and sample loss-on-ignition and carbon concentration) to
244 allow reanalysis using a standardized definition of 'peat'.

245 Estimates of peat thickness for a region are usually based on limited numbers of measurements, which
246 may be biased by over-reporting of the thickest deposits. The geometry of peatlands is such that thick
247 peats are often restricted to quite small areas (the centre of domed mires or basin-filling swamps)
248 surrounded by much more extensive areas of shallow peat. Simply taking the mean of a series of
249 measurements from the edge to the centre of a single peatland is likely to result in an overestimate of
250 the mean peat thickness. A much more robust approach is that taken by Dommain et al. (2014),
251 building on existing detailed GIS maps of Indonesian peatlands by Wahyunto et al. (2003, 2004, 2006).
252 By interpolation between field measurements, they defined GIS polygons of small areas of peat of
253 different mean thicknesses. The total volume of peat was then derived by multiplying the area of each
254 polygon by its specific thickness. This provides a useful template for future work, not least because the
255 link between field data and estimates of carbon storage is explicit, facilitating revisions as further data
256 emerge. GIS datasets can also be readily shared and assimilated into larger-scale mapping or
257 reanalysis projects, and are thus a very desirable form of research output.

258 There are few detailed studies of the geometry of individual tropical peatlands, but the exceptionally
259 detailed survey using a dense grid of 194 depth estimates across a 235 ha swamp, the CICRA peatland
260 in southern Peru (Householder et al. 2012), demonstrates that useful insights can be gained that can
261 guide more representative sampling of other peatlands in the region. In this case, comparison of the
262 peatland volume estimate derived from the entire network of points showed that less detailed
263 estimates of the volume of the same peatland based on short transects tended systematically to

264 overestimate the total peat volume. This 'calibration' was then used to adjust volume estimates based
265 on transect data only from other sites in the region.

266 Ground penetrating radar (GPR) could also be used alongside manual coring to determine peat
267 thickness and stratigraphy, and may be especially useful for studying features such as voids which can
268 be important in volume terms in forested peatlands (e.g. Slater and Reeve 2002; Parry et al. 2014).
269 There are, however, considerable practical difficulties associated with deploying relatively bulky GPR
270 over large distances in forested peatlands, though smaller instruments are currently in development.

271 Few attempts have so far been made in the tropics to derive peat thickness by remote sensing, and
272 they are exceptional cases. For example, Jaenicke et al. (2008) integrated Landsat Enhanced Thematic
273 Mapper (ETM+) and SRTM data, a network of 750 field measurements of peat thickness, and a three-
274 dimensional peatland development model to estimate the volume of domed peatlands in Kalimantan;
275 the success of this project depended on the rather special properties of the mires in question (regular
276 shape, very thick peats). Ballhorn et al. (2009) also used LiDAR measurements to estimate *changes* in
277 peat thickness over time due to burning on Indonesian peatlands. A more generally applicable method
278 for measuring peat thickness remotely is perhaps unlikely to emerge but there is scope for further
279 investigation on a site-by-site or region-by-region basis. Peat thickness can sometimes correlate with
280 other properties that are visible by remote sensing. For example, thick peats often occur towards the
281 centre of ombrotrophic peat domes. Field observations suggest that these deep, nutrient-poor peats
282 are frequently (but not always) associated with specialized and structurally distinctive vegetation
283 communities in, for example, Kalimantan (Page et al. 1999), Panama (Sjögersten et al. 2011), Peru
284 (Kelly et al. 2013; Draper et al. in press), and the Republic of Congo (G. Dargie unpublished data).
285 These plant communities are often distinctive in Landsat TM, ALOS PALSAR and other imagery.
286 However, apparent relationships between remote sensing data and peat thickness must be confirmed
287 using empirical data because many factors other than peat thickness may be equally or more
288 important in controlling vegetation composition and structure (Draper et al. in press), as is the case in
289 temperate/boreal peatlands (Wheeler and Proctor 2000).

290 *2 Bulk density*

291 Measured values of dry bulk density (DBD) for individual peat samples from Indonesia vary by almost
292 an order of magnitude, and the few available data from undisturbed sites in Amazonia vary by a factor
293 of two (all in peats with ash contents <10%; Figure 3). This variability, which arises from factors
294 including the botanical composition of the peat, consolidation of deeper peats, drainage history, and
295 measurement method, is a major source of uncertainty in estimates of the size of the peatland carbon
296 pool because calculations of carbon stocks at individual sites, or even across regions, are frequently
297 based on the mean of a very small number of DBD measurements (e.g. Page et al. 2011b; Householder
298 et al. 2012).

299 Frequently, the number of samples taken may be too small, and/or the spacing between samples may
300 be too large to capture the spatial variability of peat DBD. Within-site lateral variation in DBD has not
301 been explored systematically in tropical contexts and more work is needed to establish whether there
302 are any predictable patterns. Sometimes the stratigraphic profile of DBD is quite consistent at sites
303 within a region (Hooijer et al. 2012), but DBD can also vary systematically between peatland types
304 within a region (e.g. floodplain peatlands, domed peatlands; Shimada et al. 2001). The converse has
305 been shown in boreal peatlands (i.e. DBD varies between regions within the same peatland type;
306 Sheng et al. 2004; Yu 2012; there are insufficient data to know if this also applies in the tropics). Site-
307 specific measurements are therefore always desirable, and in general more data are needed to
308 determine whether DBD varies spatially in a predictable way.

309 A further complication is that tropical peatlands can show considerable stratigraphic variation in DBD
310 (Figure 4) due to fluvial mineral inputs (Lähteenoja et al. 2009b), long-term vegetation succession and
311 related variations in peat structure (Phillips et al. 1997; Roucoux et al. 2013), peat decomposition,
312 post-drainage consolidation (Hooijer et al. 2012), and water- or gas-filled voids. This stratigraphic
313 variation can only be addressed through field measurements and ample down-core sampling. A
314 greater palaeobotanical insight into the origins of variation in DBD in tropical peats would also be a
315 useful line of research.

316 A second potential source of error in DBD estimation is that peat samples of known volume must be
317 recovered, which is difficult to achieve reliably. One method for collecting volumetric samples is to dig
318 a pit into the peat and extract a monolith from the pit wall (Hooijer et al. 2012; Couwenberg and
319 Hooijer 2013), but this may entail continuously pumping water out of the pit which can be impractical,
320 is limited to shallow sections, and, by analogy with what is known of the effects of seasonal changes in
321 water table on peat volume in undisturbed peats (Price 2003), may lead to compaction and over-
322 estimation of DBD. Various specialised corer designs have been proposed to improve the collection of
323 volumetric samples from shallow peats (Wright et al. 1984; Givelet et al. 2004; van Asselen and
324 Roosendaal 2009; see De Vleeschouwer et al. 2010 and Glaser et al. 2012 for discussion of the relative
325 merits of different devices). At present, most workers use a Russian-type corer which is suitable for
326 use in shallow and deep peats; as a side-sampling device it offers better control over sample depths
327 than a piston corer and a low risk of sample contamination than a Hiller corer, which is especially
328 important if the peat is to be radiocarbon dated (cf. Glaser et al. 2012), but can be less effective at
329 cutting through woody peats than a corer with a serrated barrel. The volume of the *in situ* peat sample
330 is usually assumed to be identical to the internal volume of the corer. In reality, core recovery is often
331 imperfect, especially in fibrous or woody peat that cannot be cut cleanly, or where the peat is
332 structurally weak and is not retained within the corer, or (in the case of piston corers) it fails to fill the
333 barrel (Wright 1991; Dommain et al. 2011; Lahteenoja et al. 2013). Thus, DBD measurements are
334 probably often subject to large errors stemming from erroneous volume estimations. Russian-type
335 corers in particular may yield systematic underestimates of DBD (Clymo 1983). In general, large-
336 volume corers (including wide-diameter Russian corers) are to be preferred over smaller devices
337 because they will more likely retrieve a representative sample, but they can be impossible to use in
338 stiff or woody peats and are more logistically problematic compared with a smaller, lighter device. As
339 yet, there have been few systematic comparisons of different methods to assess the extent of the
340 uncertainty in volume measurements (Pitkanen et al. 2011), especially in fibrous and woody peats.
341 More research on this topic could help to quantify and, perhaps, compensate for any differences
342 between datasets that are due to the use of different sampling devices.

343 A third way in which DBD measurements made by different research groups may vary stems from
344 variation in laboratory methods, for example in the temperature at which the peat samples are dried.
345 Chambers et al. (2011) proposed a protocol for measuring DBD and other basic variables (including
346 drying at 100°C) which we recommend and which, if followed, will minimize this uncertainty.

347 *3 Carbon concentration*

348 Two principal approaches are used to estimate carbon concentration in peats. The more accurate and
349 direct technique is elemental analysis (Nelson et al. 1996; Chambers et al. 2011). In peats with low ash
350 content the carbon concentration calculated by this method typically varies between about 52 and
351 58% (Figure 5). However, many workers use mass loss-on-ignition (LOI; Heiri et al. 2001) as a cost-
352 effective way to estimate organic matter concentration. The LOI at (typically) 450°C is assumed to be
353 attributable to combustion of organic material; the remainder, the 'ash', is typically composed of
354 sedimentary mineral material and biogenic silica. Carbon concentration can then be estimated by
355 assuming that the organic material contains (e.g.) 50 wt% C (Turunen et al. 2002).

356 On the basis of available data, applying the LOI-based approach described by Turunen et al. (2002)
357 could apparently systematically underestimate carbon concentration by c. 8 wt% in tropical peats
358 with very low ash contents (Figure 6). This disparity is principally attributable to the varying
359 abundance of carbon-rich compounds, such as lignin or charcoal, in the organic matter fraction of peat.
360 However, at least at some sites, the paired measurements show a strong linear relationship, albeit with
361 some scatter. There may even be a strong linear relationship between DBD and carbon density (the
362 product of carbon concentration and DBD, measured in e.g. kg C m⁻³), sufficient to support the
363 suggestion that DBD measurements alone may be sufficient for a first-order estimate of carbon
364 concentration (Warren et al. 2012; Farmer et al. 2013). However, the strength of this relationship
365 varies and it has not been tested for tropical peats outside a few locations in Southeast Asia.

366 Therefore, for inventory purposes, and especially when undertaking work for the first time at a new
367 site, in our view it remains important to measure carbon concentration as accurately as possible, i.e.
368 using an elemental analyser following Chambers et al. (2011). LOI remains a useful tool in its own

369 right because it provides a direct measure of the organic matter content of the peat, which is valuable
370 in understanding the developmental history of a site. As with DBD, systematic studies of within- and
371 between-site variation in carbon concentration are lacking in tropical contexts.

372 The question of the number of samples to take, both horizontally and vertically, for DBD and carbon
373 concentration in a peatland is inevitably constrained by the resources available for a given project. One
374 approach to designing a sampling strategy is to assess the way in which the variation in different
375 measurements (DBD, carbon concentration, peat thickness) contributes to the overall uncertainty in
376 the estimate of the carbon pool. For example, peat thickness is usually much more variable across a
377 region than DBD or carbon concentration, suggesting that a rational use of research effort would be to
378 focus on measuring peat thickness. Resampling techniques can be used to estimate the confidence
379 interval around an estimate of the regional carbon pool for a given region (Manly 2007; Draper et al. in
380 press). Chimner et al. (2014) discussed the relative merits of different core sub-sampling approaches
381 in terms of attempting to encompass stratigraphic variation in DBD and carbon concentration in
382 Canadian peats and found that, in Canada, (a) several different approaches gave similar results and (b)
383 analysis of the DBD and carbon concentration of a single core section, from 25–75 cm depth at each
384 site, gave estimates of the total peat carbon stock that were within 15% of estimates based on
385 exhaustive sampling of entire cores from the same sites, suggesting that even a single (admittedly
386 large) sample from each core site may be adequate for inventory purposes. This conclusion must be
387 tested before being applied in other regions where, for example, frequent admixture of clay in deeper
388 peats may give very different results. In general we would recommend a more conservative approach,
389 taking several discrete subsamples throughout the full thickness of the peat (the “intermittent peat
390 sampling method” described by Chimner et al. 2014).

391 **4 Biomass**

392 A widely-used set of standard protocols has been developed for measuring above-ground biomass
393 (AGB) in *terra firme* tropical forests (e.g. Phillips et al. 2009). These protocols are, with modification,
394 applicable in forested peatlands. They should be used wherever possible because using the same
395 protocols on peat and *terra firme* enables biomass, productivity, diversity, and other key vegetation

396 parameters to be compared, which means that peatland vegetation can be understood in the broader
397 context of tropical vegetation as a whole, and can be integrated into regional assessments of AGB
398 across all ecosystems.

399 Modifications, or rather additions, to *terra firme* protocols for use on peatlands are necessary because
400 peatland vegetation is frequently dominated by plants that are usually regarded as negligible in a
401 standard forest census, for example, thin-trunked trees with diameter at breast height (DBH) < 10 cm
402 (the cut-off used in most AGB inventories), or grasses and sedges. An example of a modified protocol
403 that remains compatible with standard RAINFOR protocols is provided by Roucoux et al. (2013) at
404 Quistococha, Peru: a nested sampling design was used to record small trees with DBH between 2.5 and
405 10 cm in a series of sub-plots within their main census plot.

406 Another consideration is that peatland forests are often dominated by species (such as trunkless
407 palms) to which standard allometric equations for calculating biomass do not apply (e.g. Chave et al.
408 2005; Feldpausch et al. 2011; Gehring et al. 2011). Outside Southeast Asia, the allometric equations
409 used for AGB calculation of tropical peat swamp forests in recent literature (e.g. Kronseder et al. 2012;
410 Englhart et al. 2013) were originally developed for “moist tropical forests” in general (Chave et al.
411 2005), and the performance of these models in the distinctive pole and palm forests often found on
412 peatlands needs to be tested. Recently, species-specific equations have been developed for the most
413 important peatland palms in South America including *Mauritia flexuosa* and *Mauritiella armata*
414 (Goodman et al. 2013). Similar work remains to be carried out on many other species. The long history
415 of economic exploitation of peat swamp forest in Southeast Asia means that allometric equations there
416 are better established (Krisnawati et al. 2012).

417 New tools for assessing AGB on large spatial scales by remote sensing of canopy structure are rapidly
418 developing (e.g. in Southeast Asian peatlands: Jubanski et al. 2013; Kronseder et al. 2012). Airborne
419 LiDAR (Asner et al. 2013) and satellite-based LiDAR and L-band radar (e.g. Saatchi et al. 2011;
420 Ballhorn et al. 2011; Mitchard et al. 2012; Baccini et al. 2012) have been widely used to estimate AGB,
421 mainly in *terra firme* forests. As with mapping, more ground-reference data are needed to allow these
422 approaches to be applied confidently to peatland forest AGB assessments (Table 1).

Below-ground biomass (BGB: living roots, as opposed to the necromass or dead material contributing to peat) has hardly been studied in tropical peats, but existing research shows that root inputs can be more important to peat accumulation than leaf and stem litter, i.e. these are ‘replacement peats’ (Brady 1997; Chimner and Ewel 2005; Dommain et al. 2011). Measuring BGB is fraught with methodological problems even in *terra firme* forests, but it would be of great interest to know (in the context of understanding carbon balance and rates of carbon sequestration – see below) how much of the peat at various depths is made up of live or recently dead root material. However, for the purposes of estimating the size of carbon pools, the live and dead components of the peat do not need to be separated explicitly.

IV Carbon fluxes

In peatlands, carbon enters the peat in the form of litter and leaves it as dissolved organic carbon (DOC), particulate organic carbon (POC), and as the greenhouse gases CH₄ and CO₂. Comprehensive carbon flux measurements have been made for several well-studied northern peatlands (Roulet et al. 2007; Nilsson et al. 2008; Koehler et al. 2011), but a complete carbon budget has only been attempted once in tropical peat swamp forest (Chimner and Ewel 2005). The limited available data suggest that tropical peatland carbon sequestration rates are towards the upper end of the range for peatlands globally (Mitsch et al. 2010; Dommain et al. 2011; Glaser et al. 2012) due to a combination of high net primary production (NPP) and low decomposition rates (Dommain et al. 2011; Sjögersten et al. 2014).

The annual change in organic carbon for a peatland (ΔC_{org}) can be expressed as follows, following Roulet et al. (2007):

$$\Delta C_{org} = NPP - F_{CO_2} - F_{CH_4} - netDOC_{EX} - netPOC_{EX} \quad (2)$$

where ΔC_{org} is equivalent net primary production (NPP) minus F_{CO_2} (the gaseous flux of CO₂), F_{CH_4} (the gaseous flux of CH₄), $netDOC_{EX}$ (the waterborne DOC flux), and $netPOC_{EX}$ (the waterborne POC flux).

447 None of the quantities on the right hand side of equation 2 are easy to measure in any peatland, but in
448 the tropics, NPP is especially difficult to quantify due to the large size of the plants on forested
449 peatlands. The carbon flux can perhaps more easily be estimated using (i) the observed change in peat
450 surface height, caused by accumulation and/or subsidence, relative to a fixed stake (e.g. Nagano et al.
451 2013; Couwenberg and Hooijer 2013); (ii) dating the basal peat to establish the apparent long term
452 rate of carbon accumulation (LORCA: Clymo et al. 1998; Turunen et al. 2002; 'long term' in this context
453 means centuries to millennia) as a direct measure of carbon sequestration over a given period of time
454 (although this can differ substantially from present rates of carbon accumulation: Joosten and Clarke
455 2002:34); or (iii) measured rates of litter production and carbon losses through decomposition
456 (Chimner and Ewel 2005). Our discussion below focuses on the prospects for the direct measurement
457 of litter input and decomposition, greenhouse gas fluxes (chiefly CO₂ and CH₄), and waterborne carbon
458 fluxes, all of which are needed both to quantify the carbon flux in tropical peatlands and in order to
459 develop a fuller mechanistic understanding of the controls on their carbon balance and greenhouse gas
460 fluxes.

461 *1 Litter inputs*

462 The transformation of different types of litter in tropical forested peatlands – roots, non-woody leaf
463 and stem litter, and woody debris ranging from twigs and small branches to whole tree trunks – into
464 peat is poorly understood (Tie and Esterle 1992; Brady 1997; Sulistiyanto 2004; Chimner and Ewel
465 2005; Shimamura and Momose 2005). The quantity of litter inputs and the rate at which each litter
466 type decomposes determine their contribution to the peat carbon pool. Leaves can comprise the bulk
467 of above-ground litterfall in peat swamp forests (Sulistiyanto 2004) but they typically decompose
468 much more rapidly than woody debris and roots and therefore contribute little to the overall
469 accumulation of peat (Chimner and Ewel 2005), except perhaps where leaves accumulate in ponds on
470 the peat surface (Gastaldo and Staub 1999).

471 Above-ground litterfall can be collected and weighed using nets of e.g. 1×1 m, depending on the size of
472 the litter, although the logistics of the necessarily frequent sampling (due to high litter decomposition
473 rates in the traps) can be restrictive. Suitable protocols have been developed by the CTFS Global Forest

474 Carbon Research Initiative (www.ctfs.si.edu) and Harrison (2013); comparable sampling schemes
475 have also been used in lowland tropical rainforest (Chambers et al. 2000; Nepstad et al. 2002). The
476 contribution of large woody debris is harder to measure, and is almost unexplored in tropical
477 peatlands. The frequency and relative importance of branch- and tree-fall events can be estimated by
478 repeated litter surveys along transects (Waddell 2002) or in census plots (Chimner and Ewel 2005;
479 Woodall and Monleon 2008; Baker and Chao 2011), although no standard method has yet been agreed
480 upon (Larjavaara and Muller-Landau 2011); further methodological research is needed.

481 Equally, very few data on root dynamics are available for tropical peatlands. Root growth, especially of
482 fine roots, can be measured using the ingrowth core or bag method (e.g. Symbula and Day 1988; Neill
483 1992; Brady 1997; Finér and Laine 1998; Metcalfe et al. 2008), though this can be problematic:
484 important considerations are the linearity of root growth over the study period and temporal variation
485 in root production, e.g. between dry and wet seasons (Metcalfe et al. 2008), as well as changes in soil
486 structure caused by the removal of roots from the soils during preparation of the ingrowth core, which
487 may affect later root growth. Root mortality (i.e. root litter input to the necromass) is extremely
488 difficult to determine and involves differentiating between live and dead root material (Finér and
489 Laine 1998). Alternative approaches, such as the use of minirhizotrons which allow *in situ*
490 measurements of root growth and mortality (Iversen et al. 2012), have been used successfully in
491 lowland tropical rainforest (Metcalfe et al. 2007) but not yet in tropical peatlands.

492 *2 Litter decomposition*

493 Potential *in situ* litter decomposition rates in tropical peatlands appear higher than in
494 temperate/boreal peatlands (Brady 1997; Chimner and Ewel 2005; Yule and Gomez 2008),
495 presumably due mainly to the year-round higher ambient temperatures, although other factors
496 including litter composition, water table depth, and pH may also be involved (Qualls and Haines 1990;
497 Chimner and Ewel 2005; Yule and Gomez 2008). Few systematic studies have been carried out,
498 especially outside Southeast Asia.

499 Standard techniques for measuring the decomposition rate of fine litter are readily applied in tropical

peatlands, in which rapid decomposition allows meaningful results to be obtained over sampling periods as short as two years (Yule and Gomez, 2008; Hoyos 2014). Mesh bags containing known dry weights of litter are firmly anchored to the ground to prevent them being washed away during floods, or are buried in the peat and collected at regular intervals to calculate weight loss (Chimner and Ewel 2005; Wright et al. 2013a; Hoyos 2014). Decomposition of large woody debris such as fallen trees or large branches is also presumably important in forested peatlands, but sampling is more difficult, usually requiring repeated surveys over long intervals, and data are lacking. One approach would be to follow (or build on) RAINFOR protocols to establish permanent forest census plots, which include a protocol for measuring coarse woody debris (Baker and Chao 2011). Census plots that are part of a network such as RAINFOR, where the data are relevant to many research questions, are more likely than not to be revisited over many years and hence to generate the necessary long-term datasets.

3 Greenhouse gas fluxes

To date, most data on GHG fluxes from tropical peatlands have been collected during daylight hours using static sampling chambers placed on the peat surface (Jauhiainen et al. 2005, 2008, 2012; Melling et al. 2005a, b; Sjögersten et al. 2011; Wright et al. 2011, 2013b). Automated sampling has not yet been widely adopted but is becoming more common (Sundari et al. 2012; Hirano et al. 2014). Measured GHG fluxes from peat swamp forests vary greatly both across the tropics (Sjögersten et al. 2014) and within sites (Wright et al. 2013b), often correlating with mean annual water table depth (Couwenberg et al. 2010).

Substantial temporal variation (diurnal and seasonal) poses a major challenge to obtaining reliable estimates of GHG emissions. Current data suggest that temporal variation exceeds the spatial variation between forest types (Wright et al. 2013b), and that there is frequently a strong correlation between GHG efflux and temporal variation in water table depth (Jauhiainen et al. 2005; Hirano et al. 2009, 2014; Sundari et al. 2012). Long-term data (e.g. over more than a year) obtained at regular (e.g. monthly) intervals are scarce, so the magnitude of intra- and inter-annual variation in fluxes is unclear. Strong diurnal variation in CO₂ and CH₄ efflux (e.g. Hirano et al, 2009; Wright et al. 2013b; Hoyos 2014) means that measurements intended for estimating net GHG fluxes need to include

527 measurements at 2–4 hourly intervals or better. The effect of weather (e.g. rainfall events) and
528 seasonality (e.g. dry/wet season contrasts) on GHG emissions, and the importance of ebullition in
529 methane emission, also remain to be thoroughly investigated. Automated samplers capable of frequent
530 (sub-hourly) measurements over long periods (many months; e.g. Goodrich et al. 2011) can help to
531 resolve these issues.

532 Alternatively, the eddy-flux correlation approach can be used to acquire gas flux data with high
533 temporal resolution, spatially integrated over hundreds of metres. Eddy-flux systems are currently
534 under-used in tropical peatlands, with data presently only available from sites in Narathiwat Province,
535 Thailand (Suzuki et al. 1999) and Kalimantan, Indonesia (Hirano et al. 2009, 2012). Additional systems
536 will shortly become operational in Brunei, Sarawak (Malaysia), and Peru.

537 Additional uncertainty in estimates of peat decay rate arise because most CO₂ flux data from tropical
538 peatlands do not separate autotrophic (from roots) and heterotrophic respiration (from decomposing
539 peat) which makes it difficult to use peat surface measurements of GHGs to assess peat decay rates
540 (Page et al. 2011a). In plantations with regularly spaced trees and little ground cover, attempts have
541 been made to distinguish between soil and root respiration by measuring CO₂ release within and
542 between rows of trees (Jauhiainen et al. 2012). Alternative approaches to separating autotrophic and
543 heterotrophic respiration are (i) to compare the CO₂ release from buried mesh collars which restrict
544 in-growth of roots, with collars which allow roots to grow in (Nottingham et al. 2011), or (ii) using a
545 trenching approach (Mäkiränta et al. 2008), which involves isolating a patch of ground from root
546 influences by cutting/digging through the roots around the plot (Mäkiränta et al. 2008), although this
547 may affect the peat moisture status in trenched plots, with implications for CO₂ fluxes.

548 A further uncertainty regarding net GHG emissions is related to the pathway for CH₄ emissions
549 through tree stems, which a study by Pangala et al. (2013) at a peatland in Borneo found to account for
550 a very large proportion (62–87%) of ecosystem CH₄ emissions. The extent to which this pathway is
551 generally important needs to be established by comparable studies at other sites.

552 4 Waterborne carbon fluxes

553 Waterborne carbon fluxes (DOC and POC) from tropical peatlands represent a major source of
554 uncertainty in their overall carbon balance. In temperate peatlands, waterborne carbon typically
555 accounts for c. 10% of total carbon export (Limpens et al. 2008). In the few studies in which DOC and
556 POC fluxes from tropical peatlands (all in Southeast Asia) have been measured (Yoshioka et al. 2002;
557 Baum et al. 2007), they have been found to be approximately double those from temperate peatlands
558 (IPCC 2014). Moore et al. (2011) estimated that the peat-covered part of Indonesia alone was
559 responsible for 10% of global fluvial DOC export to the ocean. Waterborne fluxes may be especially
560 significant in degraded peatlands where the forest vegetation has been removed and the peat has
561 destabilised (cf. Moore et al. 2013), and in floodplain peatlands where fluvial erosion can remove large
562 blocks of peat *en masse* during floods. Further quantification of these processes is needed.

563 Total export of waterborne carbon can most easily be estimated for peatlands which have a clear
564 hydrological boundary and discrete outflows, by measuring DOC and POC concentrations in drainage
565 streams regularly during annual or longer periods, along with the total water discharge (Billett et al.
566 2004; Moore et al. 2011, 2013). The achievable temporal resolution of measurements is a critical
567 limitation. Woody tropical peats often have high saturated hydraulic conductivity in their near-surface
568 layers, but below-ground flow is frequently insufficient to shed the large and sporadic inputs from
569 rainfall (Kelly et al. 2013); evapotranspiration and, especially in the wet season, surface runoff play a
570 large role in the hydrological budget. The pathway taken by water as it leaves a peatland affects its
571 DOC and POC load. Water which is shed rapidly through surface runoff may have a low DOC
572 concentration due to its shorter residence time, but equally, especially in degraded peatlands, rapid
573 runoff may cause peatland erosion and carry a greater POC load. In such hydrologically dynamic
574 peatlands, discrete pulses of DOC and POC losses may be missed unless monitoring is carried out very
575 frequently. In a recent study, Moore et al. (2013) focused sampling effort on the peak of the dry season
576 and wet season, taking measurements every week during these periods. For the rest of the year, they
577 took samples every fortnight. We recommend this as a minimum sampling resolution for future
578 studies, taking into account the difficulty of regular sampling in many tropical peatlands; where

sampling can be undertaken more intensively (for example, using automated samplers) then this should be attempted. Further research on tropical peatland hydrology (e.g. Kelly et al. 2014) leading to more reliable hydrological models would also help in estimating waterborne carbon fluxes.

V Conclusions

Tropical peatland research with a focus on their role as carbon stores, sinks and sources is becoming an increasingly active field, and an important one in relation to management of the global carbon cycle. In this review we have identified many research needs, including methodological problems, and have suggested some approaches to tackling them. In our view, however, the overarching need is for a more coordinated approach to data collection and sharing. This is necessary to allow us to address the most fundamental, large-scale questions about how much peat exists in the tropics, and where it is; and about the role of tropical peatlands in the global carbon cycle, today and in the future. Our main conclusions and recommendations are as follows:

1. Tropical peatland research would benefit from a network of sites where basic measurements have been made using identical methodologies. A precedent exists in the well-organised, extensive networks of permanent tropical forest census plots (e.g. RAINFOR and AFRITRON). Where practical, methods should be compatible with those used for peatland research outside the tropics, but perhaps more importantly, they should be compatible with methods used in other tropical ecosystems (particularly *terra firme* forest), in recognition of the fact that effective management depends primarily on being able to compare the relative costs and benefits of managing peatlands and other ecosystems in the same region for carbon storage and other ecosystem services. Table 1 proposes a set of measurements which are cheap and practical to implement as part of basic site description, and which would help to build a pan-tropical dataset that would put regional and global estimates of tropical peatland carbon stocks and fluxes on a firmer footing.
2. Concerted effort to focus research on particular sites, drawing on both the social and the natural sciences, has proven successful at one tropical peatland (Allen et al. 2005; Chimner and Ewel, 2005; Drew et al. 2005). We would like to see the research community continue to build on this collaborative

605 and interdisciplinary approach by establishing a small number of keystone sites where a rich body of
606 knowledge can be accumulated over time. This would facilitate the testing of conceptual and numerical
607 models of peatland processes, and would help to build long-term datasets that can be used to analyse
608 temporal variability in peatland behaviour.

609 3. As in many other fields, it would be helpful if data were routinely published in full, in tabular form in
610 papers (as supplementary data if necessary) or in appropriate data repositories such as the Carbon
611 Dioxide Information Analysis Center (<http://cdiac.ornl.gov/>) or the UK Environmental Information
612 Data Centre (<http://www.ceh.ac.uk/data>), in order to facilitate reanalysis and synthesis. Full
613 publication of data is increasingly required by grant funding bodies. The fact that this so rarely
614 happens at present suggests that mechanisms are needed to incentivise sharing of data between
615 researchers. Again, existing networks such as RAINFOR (Malhi et al. 2002) offer precedents to follow,
616 in terms of 'ground rules' that incentivise data sharing by guaranteeing opportunities for co-
617 authorship of any publications that result.

618 4. There is, separately, a need for a community-wide data synthesis project to build a GIS-compatible
619 database on carbon storage in tropical peatlands (and indeed, peatlands globally) that would facilitate
620 inter-site comparisons.

621 5. Throughout this review we have identified research priorities which, if addressed, would improve
622 our ability to make reliable measurements and to extrapolate from point measurements to regional
623 and global assessments of peatland carbon stocks and fluxes. These include:

- 624 a. Studying the relationships between peat properties, the overlying vegetation, and their remote
625 sensing signatures;
- 626 b. Developing radar/LiDAR techniques for mapping tropical peatlands and measuring AGB;
- 627 c. Investigating the use of multiple remote sensing methods in combination in mapping tropical
628 peatlands;
- 629 d. Collaboratively developing large ground reference point datasets to support remote sensing;
- 630 e. Investigating further the potential for inferring peat thickness by GPR and remote sensing;

- f. Systematically comparing different volumetric peat sampling methods;
- g. Investigating the spatial and stratigraphic variation in peat DBD and carbon concentration;
- h. Investigating the relative importance of different litter inputs (including coarse woody debris and roots) to peat formation/C flux;
- i. Improving our understanding of the spatial and, especially, temporal variation in greenhouse gas fluxes from peatlands;
- j. Investigating further the importance of vegetation (especially trees) as conduits for greenhouse gases in tropical peatlands.

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1036 **Table**

Variable	Method(s) and key references
Location	± 10 m precision of good quality consumer-grade handheld GPS or GPS/GLONASS units is adequate.
Peat thickness	Measurement by coring or augering, taking care to define the base of the ‘peat’ using reproducible criteria, i.e. taking ample samples for carbon concentration and loss-on-

	ignition measurements: e.g. Parry et al. (2014).
Peat carbon concentration	Measured by elemental analysis, including samples from the full range of peat depths: Chambers et al. (2011); Chimner et al. (2014).
Peat dry bulk density	Each carbon concentration measurement should have an associated dry bulk density measurement: see Chambers et al. (2011).
Canopy height	Height of the ten tallest trees within 20 m of the core site, measured using a clinometer and tape measure, or a laser rangefinder: Phillips et al. (2009).
Vegetation composition/structure	Ideally, installation of a permanent 0.5–1 ha vegetation sampling plot following RAINFOR protocols (Malhi et al. 2002), extended where appropriate (e.g. to include small trees, shrubs and herbs where these are important, and coarse woody debris). Where this is impractical, a general description of the vegetation structure and dominant species within 20 m of the core site is sufficient for most remote sensing studies.

Table 1. A suggested protocol for site description which facilitates basic data comparison, and development/testing of remote sensing techniques for peatland mapping and characterization.

Figure captions

Figure 1. Distribution of the peat carbon pool in the tropics, based on country-scale estimates from Page et al. (2011b; dotted regions indicate no data). In Australia, the estimate refers to the state of Queensland only. Examples of lowland peatlands discussed in the text are indicated as follows: SSPS:

1043 San San Pond Sak, Panama; PMF: Pastaza-Marañón Foreland, Peru; CC: Cuvette Centrale, Republic of
1044 Congo/Democratic Republic of Congo; TB: Tasek Bera, Malaysia; K: Kalimantan, Indonesia.

1045 **Figure 2.** Vegetation classification of the Changuinola peat dome in the San San Pond Sak tropical
1046 peatland, Panama, using Landsat Thematic Mapper imagery, supported by both aerial photographs
1047 and field data as sources of reference.

1048 **Figure 3.** Dry bulk density (DBD) values from published peat sequences. The box plots show the range
1049 of the data (dashed bars) and the lower, middle and upper quartiles (horizontal lines); the width of the
1050 bars is proportional to the square root of the size of each dataset (the total number of samples is 90);
1051 outliers are shown as circles. Only data from peats with <10% ash are shown. Note that the Sebangau
1052 peatland is in Indonesia; all other peatlands are from the Peruvian Amazon. Data sources: Wüst et al.
1053 2002, 2003; Page et al. 2004; Lähteenoja et al. 2009a; Lähteenoja and Page, 2011.

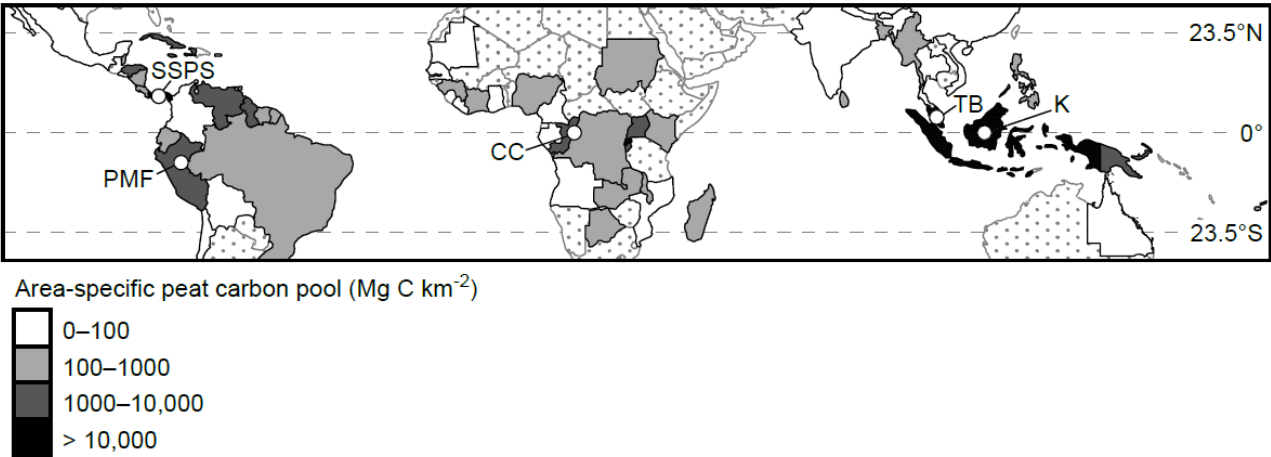
1054 **Figure 4.** Dry bulk density and ash content from core SA6.5, Kalimantan, Indonesia (Page et al. 2004),
1055 illustrating stratigraphic variation in DBD values.

1056 **Figure 5.** Organic carbon values from published peat sequences (references and symbols as for fig. 2).
1057 Only data from peats with <10% ash are shown. Note that all records are from the Peruvian Amazon,
1058 except Tasek Bera (Malaysia) and Sebangau (Indonesia). Data sources: Wüst et al. 2002, 2003; Page et
1059 al. 2004; Lähteenoja et al. 2009; Lähteenoja and Page, 2011.

1060 **Figure 6.** Carbon density measured using an elemental analyser plotted against ash content
1061 determined by loss-on-ignition (LOI) for some tropical peats. The straight line indicates the
1062 relationship used by Turunen et al. (2002) to estimate carbon content from LOI data. Only data from
1063 peats with <10% ash are shown. Data sources: Wüst et al. 2002, 2003; Page et al. 2004; Lähteenoja et
1064 al. 2009a; Lähteenoja and Page, 2011.

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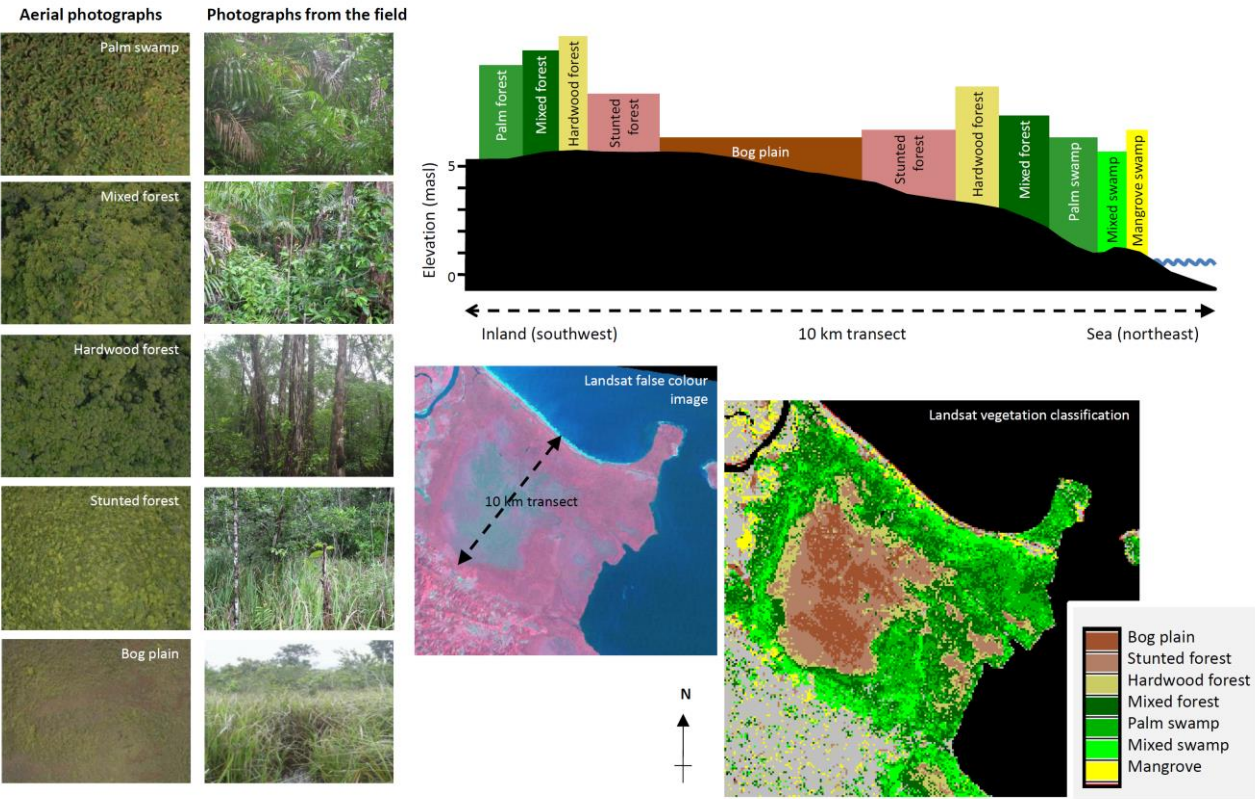
1066 Fig. 1



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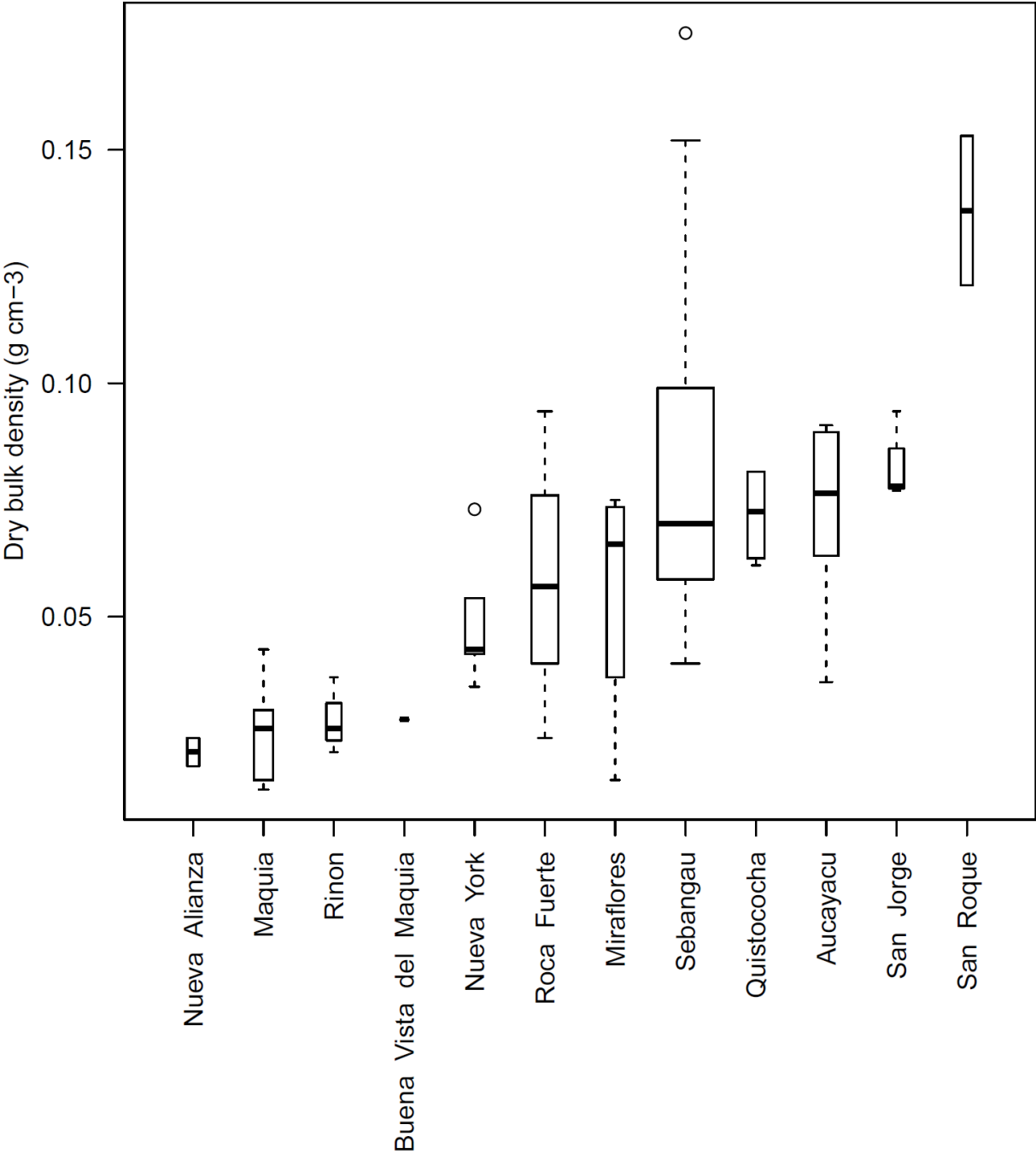
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1069 Fig. 2



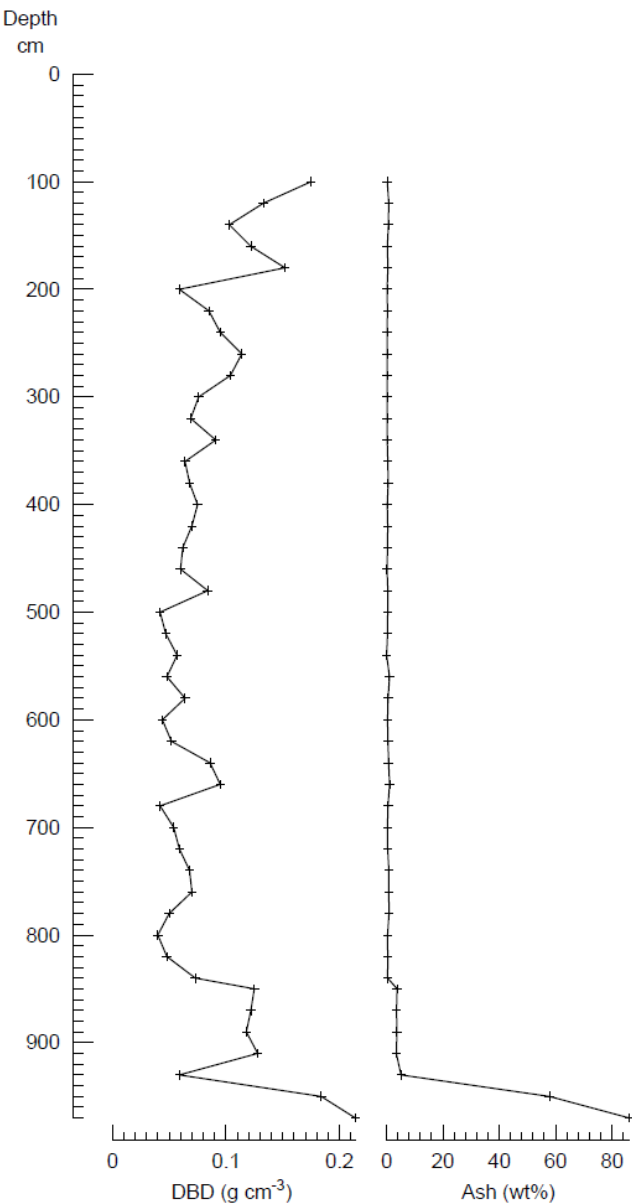
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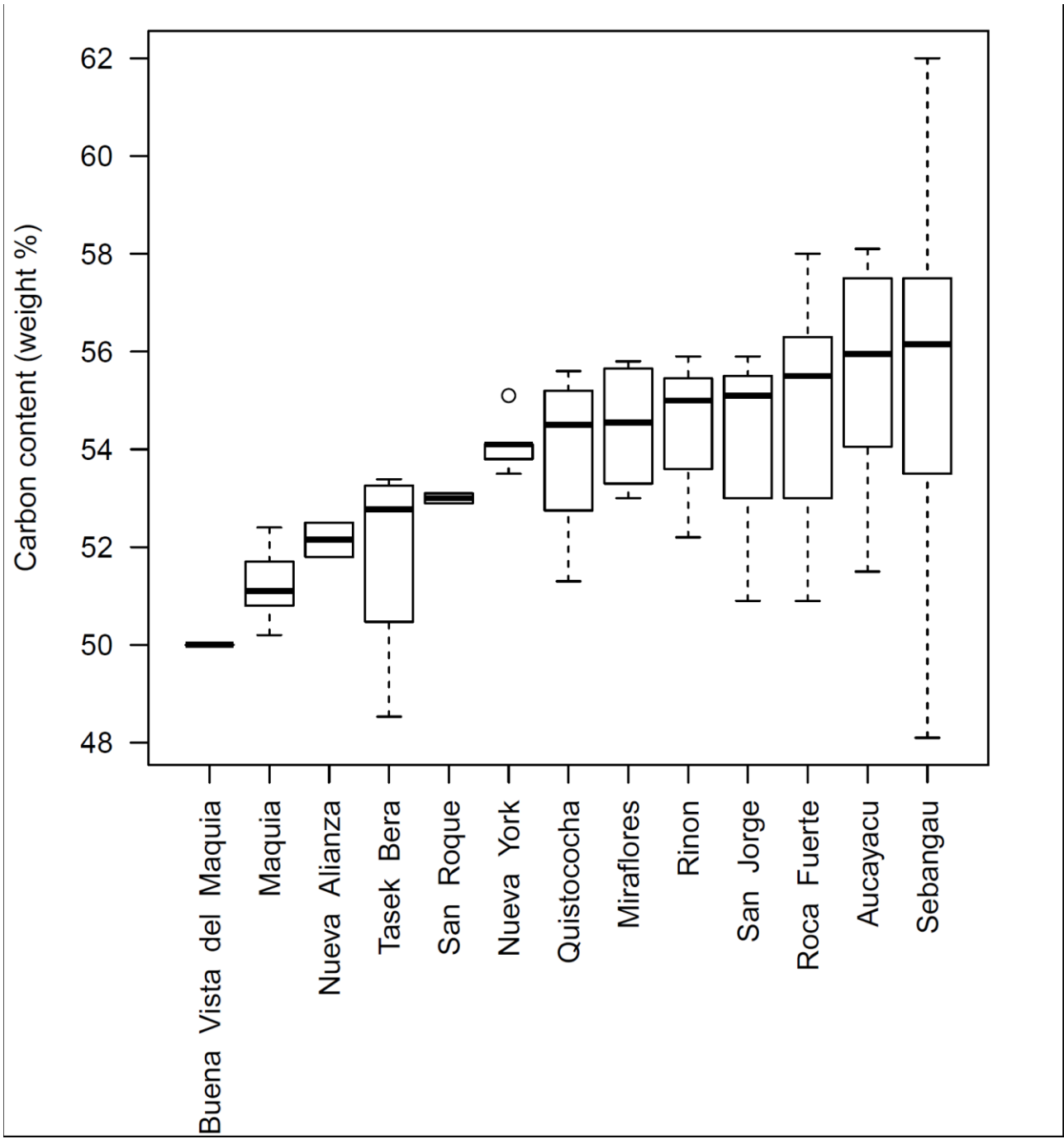
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